UNITED STATES
NUCLEAR REGULATORY COMMISSION
WASHINGTON, D. C. 20555

AUG 20 1980

Docket No. 50-320

MEMORANDUM FOR: Paul Leech
TMI Program Office

THRU: James P. Knight, Assistant Director
for Components and Structures Engineering,
Division of Engineering

FROM: George E. Lear, Chief
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SUBJECT: ASSESSMENT OF GROUNDWATER LIQUID PATHWAY FROM LEAKAGE OF THREE MILE ISLAND CONTAINMENT

The Hydrologic Engineering Section has evaluated the liquid pathway considerations of leakage into the ground of the contaminated sump water within the TMI Unit 2 containment. Using conservative methods of analysis, we estimate that the levels of radioactivity in drinking water due to the plant would be no more than 4.5×10^{-8} microcuries/ml for SR-90, 2.9×10^{-10} microcuries/ml for Cs-137 and 2.7×10^{-7} microcuries/ml for tritium. All these levels are well below MPC for unrestricted use. Values of concentration for the maximum individual dose rate due to fish consumption should be four times greater than the drinking water concentrations. This review was performed by Richard Codell.

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Attachment:
As stated

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HYDROLOGIC ENGINEERING SECTION, HGEB

ASSESSMENT OF GROUNDWATER LIQUID PATHWAY FROM LEAKAGE OF CONTAINMENT WATER AT THREE MILE ISLAND, UNIT 2

by Richard B. Codell

The staff has conservatively estimated the concentrations in the Susquehanna River which would result from the leakage of radioactive water in the containment building.

The Unit 2 containment has approximately 700,000 gallons of radioactively contaminated water on the floor. Major quantities of tritium (0.95 microcuries/ml), Cs-137 (163 microcuries/ml), Cs-134 (24 microcuries/ml) and Sr-90 (2.61 microcuries/ml) are dissolved in this water. The postulated accident is a breach of containment which allows part or all of this contaminated water to escape to the ground.

The reactor building water level was 290 feet MSL in April 1980 as shown in figure 1. The corresponding groundwater level was 285 feet MSL. Under these circumstances, the maximum water loss would be less than 2/3 of the 700,000 gallons. The staff conservatively assumed that 100% of the water was lost to the groundwater.

Release Mechanism

There is no realistic mechanism which would release large quantities of the radioactive water to the environment. For the purposes of this study, it was

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arbitrarily and conservatively assumed that water released from the containment would seep into the ground over an area equal to that of the containment building floor. The range of permeability values for the surficial soils was 10^{-2} to 10^{-3} cm/sec (FSAR). The staff therefore conservatively estimated that it would take from 0.25 to about 2.5 days for all the water in the containment to seep away. The natural water table has a measured slope in the direction of the river of about 0.006 (FSAR). The staff estimated that the travel time under normally occurring water table conditions would range from 350 days to 7060 days. The true groundwater travel time is almost certainly longer than the minimum however, for the following reasons:

1. The actual gradient is probably smaller than that reported during construction of the site because much of the natural land surface is now covered with impermeable concrete and is also well drained; and

2. A travel time of 350 days would require a recharge to the water table of about 50% of all rain falling onto the land. Typical groundwater recharge would be only a few percent of the rainfall after evapotranspiration and runoff in a well graded area.

The released contaminated water would recharge the water table creating a local groundwater "mound" which would perturb normal flow. The effect of the 700,000 gallon volume of contaminated water on the water table will be substantial, but of short duration. It is possible to demonstrate by way of an example that the perturbation of flow in the water table can be neglected for the most important radionuclides, Sr-90 and Cs-137, which have large liquid pathway dose factors and relatively long half lives.

Because of sorption of radionuclides on the soil, transport of sorbed radionuclides from the reactor containment to the river will take a minimum of tens of years. In a ten year period, the quantity of groundwater flowing directly under the containment under the assumptions used to calculate 350 day groundwater time would be approximately 1.5×10^7 gallons, as compared to the 0.7×10^6 gallons in the containment. The water in the containment could therefore, provide only a few percent, at most, of the water necessary to transport sorbed radionuclides to the river. Radionuclides which are not sorbed, notably tritium, will be significantly affected by the perturbation in groundwater flow, however.

Transport in the groundwater will take place through the unconsolidated material and weathered bedrock underlying the site. The unconsolidated material is sand, silt and gravel. The bedrock is red siltstone. There were apparently no chemical analyses performed on these materials which would indicate to what extent the radionuclides in the contaminated water would be retarded. Literature values of retardation are not reliable for realistic assessments, but for lack of field data, conservative values have been chosen. Field and laboratory data have been compiled for sorption on a number of soils and rocks (Isherwood, 1977). The lowest sorption coefficient, K_d, for unconsolidated material reported is 1.4 ml/gm for strontium and 22 ml/gm for cesium in quartz sand. Values for siltstone and alluvial materials such as that at the TMI site would be expected to be much higher, but the above low values are chosen for conservatism.

The retardation coefficient, which is related to K_d, is the speed at which the groundwater moves relative to the sorbed substance, and is always greater than or equal to 1:

$$R_d = 1 + \frac{\rho K_d}{n} ,$$

where ρ is the density of the medium and n is the total porosity. If a typical density of 2.0 gm/cc and a total porosity of 0.15 are chosen, R_d is 24 for strontium and 294 for cesium.

Transport model for sorbed radionuclides

A simple transport model is proposed to calculate the flux of sorbed radioactivity to the river. Consider the conservative situation where the entire reactor building volume seeps into the water table over an area the size of the floor of the building. If the seepage occurs quickly compared to the movement of the natural water, then the concentration profile in the water table would resemble the squared pulse shown in figure 2a. As the contamination is eluted by the flowing groundwater, the front and back end of the square pulse become rounded by diffusion and dispersion as shown in figures 2b and 2c. Dispersion in unconsolidated alluvial deposits is judged to be too small to diminish the concentration in the center of the deposit significantly (the bases for this are given in Appendix A). It is, therefore, conservatively assumed that the concentration in the center of the pulse remains the same as when it was first released except that it is reduced to account for radioactive decay.

The pulse flows into the Susquehanna River at a rate determined by the groundwater flow rate, the retardation coefficient and the decay coefficient:

$$\text{Flux} = \frac{Mu}{\ell R_d} \exp\left(-\frac{\lambda x R_d}{u}\right) \text{ curies/day}$$

where M is the source term of the radionuclide, curies

ℓ is the length of the pulse, ft

u is the groundwater velocity, ft/day

R_d is the retardation coefficient

λ is the decay coefficient $\text{day}^{-1} = \frac{.693}{\text{half life in days}}$

x is the distance from the center of the source to the river.

For the present example, the parameters chosen are given below:

	Sr 90	Cs 137
M, curies	6915	4.32×10^5
R_d	24	294
$t_{1/2}$ years	29	30.1
ℓ, feet	150	150
u, ft/day	1.7	1.7
x, ft	600	600

The maximum fluxes of Sr-90 and Cs-137 calculated are 1.88 and 0.024 curies/day respectively.

Transport of Tritium

Unlike the Sr-90 and Cs-137, the flux of tritium into the Susquehanna River will depend to a degree on the flow induced in the groundwater by the large volume of the spill. The flux of tritium is estimated by assuming that the dissolved radioactivity is released to the water table instantaneously. The circle of contaminated water displaces fresh water in the water table with no mixing. The contaminated water then moves in the ambient flow of groundwater towards the Susquehanna River, carrying with it the dissolved tritium, but leaving sorbed nuclides behind. Details of this model are given in Appendix B.

The maximum flux of tritium is estimated to be 2.8×10^{-6} curies per second into the Susquehanna River.

Surface Water Dilution

The release of Sr-90 and Cs-137 to the Susquehanna River will continue over a long period if left unchecked. Since doses are usually calculated over a period of at least one year, the logical choice of a stream flow is the annual average of 34,000 cubic feet per second (CFS). All downstream drinking water users on the Susquehanna River are located far enough downstream that total mixing of the effluent across the channel would be expected. Travel time to downstream users would be negligable compared to that of the groundwater pathway. The concentrations in the Susquehanna River based on the annual average flow would be 4.5×10^{-8} microcuries/ml for Sr-90, 2.9×10^{-10} microcuries per milliliter for Cs-137, and 2.9×10^{-7} microcuries per milliter for tritium. Maximum permissible concentrations (MPC) for unrestricted drinking water from 10 CFR 20 are 3×10^{-7} microcuries per milliliter for Sr-90, 2×10^{-5} microcuries per milliliter for Cs-137 and 3×10^{-3} microcuries per milliliter for tritium. The calculated river concentrations are thus orders of magnitude below MPC.

Minor amounts of Cs-137 would be expected to become attached (sorbed) to suspended and bottom sediments, especially behind the dams on the river. This would partially cleanse the water column of Cs-137, while increasing the exposure to bottom dwelling organisms and aquatic life feeding on them. These phenomena would be expected to have only a minor effect on dose, and are neglected in the present analysis.

Staff practice for calculating the highest fish exposure is to use the average concentration within 1/4 mile of the point of release. In this case, the 1/4 mile average would be the concentration in the middle channel of the Susquehanna River since the effluent first flows into this channel. The flow in the middle channel is estimated to be about 25% of the total river flow. The concentrations for fish exposure assessments are therefore, four times greater than those used for the drinking water, namely 1.8×10^{-7} microcuries per milliliter for Sr-90, 1.16×10^{-9} microcuries per milliliter for Cs-137, and 1.2×10^{-6} microcuries per milliliter for tritium. It should be noted that these concentrations should be used for the maximum exposure case only. The previously identified drinking water concentrations should be used for fish downstream of the plant between the 1/4 mile limit, and the tidal portion of the river and the Chesapeake Bay.

Appendix A - Dispersion from an Area Source in 2 Dimensions

The concentration in an aquifer downgradient of an instantaneous nondecaying source strength of 1 unit has been shown to be:

$$C_i = \frac{1}{n_e R_d} X\ (x,t)\ Y\ (y,t)\ Z\ (z,t)$$

where X, Y, and Z are the Green's functions in the X, Y, and Z directions respectively (Codell, 1980). The Green's functions describe the spreading in their respective coordinate directions and may be viewed as being independent of each other.

The X Green's function describes spreading in the x direction, which in this case is the direction of flow for the groundwater. For an area source of length ℓ in the direction of flow as shown in figure 2:

$$X = \frac{1}{2\ell} \left\{ \text{erf} \frac{(x + \frac{\ell}{2}) - \frac{ut}{R_d}}{\sqrt{4D_x t/R_d}} - \text{erf} \frac{(x - \frac{\ell}{2}) - \frac{ut}{R_d}}{\sqrt{4D_x t/R_d}} \right\} \qquad \text{(A1)}$$

where x is the distance from the center of the source, ft

ℓ is the length of the source, ft

u is the groundwater velocity, ft/day

t is time, days

D_x is the dispersion coefficient, ft^2/day

R_d is the retardation coefficient

erf is the error function.

As the center of the pulse reaches the river in Figure 2:

$$t = \frac{xR_d}{u} \quad . \tag{A2}$$

Equation A1 reduces to:

$$X = \frac{1}{2} \text{ erf } (\text{arg}) \tag{A3}$$

$$\text{where arg} = \frac{\ell/2}{\sqrt{4D_x x/u}} \quad .$$

If we postulate that longitudinal dispersion is unimportant in diminishing the concentration in the pulse, then the functional dependence of X on D_x and t is zero or small. Such a condition is met by:

erf (arg) close to 1.0.

As a criteria for specifying that dispersion has less than a 5% effect on the concentration at the center of the pulse we may state:

$$\text{erf (arg)} \geq 0.95.$$

The error function is a monotonically increasing function which approaches unity. For arg = 1.38, erf (arg) = 0.95, or within 5% of unity. Therefore:

$$\text{arg} = \frac{\ell/2}{\sqrt{4D_x x/u}} \geq 1.38 \quad . \tag{A4}$$

The dispersion coefficient is related to velocity in the x direction:

$$D_x \approx \alpha u \ , \qquad \text{(A5)}$$

where α is the dispersivity, ft.

Therefore, eq. A4 reduces to:

$$\frac{\ell/2}{\sqrt{4\alpha x}} \geq 1.38$$

$$\text{or} \quad \frac{\ell^2}{\alpha x} > 30.5 \ . \qquad \text{(A6)}$$

A typical dispersivity in unconsolidated sand would be $\alpha = 0.3$ ft. (Perkens, 1977). For the case $\ell = 150$ ft., $x = 600$ ft. Therefore, $\frac{\ell^2}{\alpha x} = 125$, which is much greater than 30.5. Even if $\alpha = 1.0$ feet, $\frac{\ell^2}{\alpha x} = 37.5$. We can therefore, conclude that longitudinal dispersion will probably have less than a 5% effect on diminishing the maximum groundwater concentration at the TMI site.

Appendix B - Flux of Tritium from Initial Spill

The volume of water which leaks into the water table will alter the water level and induce flow of its own. This flow will carry dissolved radionuclides, especially those such as tritium which are not easily sorbed.

In an isotropic homogeneous aquifer of inifinite lateral extent and of constant thickness H (ft), the horizontal flow of groundwater can be estimated by the partial differential equation (McWhorter and Sunada, 1977).

$$\frac{\partial^2 h}{\partial x^2} + \frac{\partial^2 h}{\partial y^2} = \frac{1}{\alpha} \frac{\partial h}{\partial t} \qquad \text{(B1)}$$

where h is the piezometric level, ft,

α is the transmissivity = kH/n_e,

n_e is the effective porosity,

k is the permeability, ft/day,

t is time, days,

and x and y are the coordinates, ft.

A conservative model of this induced flow assumes that the water is instantaneously injected at a point into an infinite, continuous, isotropic medium as shown in figure 3. By analogy to the transport of heat from a line source in an infinite cylinder (Carslaw and Jaeger, 1959), the water surface elevation h can be predicted as a function of time:

$$h = \frac{Q}{n_e 4\pi\alpha t} \exp\left(-\frac{r^2}{4\alpha t}\right) \qquad \text{(B2)}$$

where Q is the quantity of water injected, ft^3,

r is the radial distance from the point moving with the fluid, $(x_0 - ut, y_0)$ where (x_0, y_0) is the point of release and u is the pore velocity

and the other terms are as previously defined.

This equation is correct only for confined aquifers where the thickness of the saturated layer does not vary. In the present situation, the aquifer is unconfined, so h is also the piezometric level. Some error will be introduced, especially for short times and distances close to the point of release.

In the reference plane of the center of the spreading mound, which is moving with the ambient groundwater pore velocity u, the flow is away from the center at radial velicity U_r:

$$U_r = -\frac{\partial h}{\partial r} k/n_e = \frac{Qkr}{8\pi n_e^2 \alpha^2 t^2} \exp(-r^2/4\alpha t) \quad . \qquad \text{B3}$$

If the conservative assumption is made that contaminated water will exactly displace fresh water in the aquifer, the radius of the circle of contaminated water will grow at the radial velocity U_r:

$$r = \int_o^t U_r dt = \int_o^t \frac{Qkr}{8\pi n_e^2 \alpha^2 t^2} \exp(-r^2/4\alpha t)\, dt \quad . \qquad \text{B4}$$

Equation B4 is solved by numerical integration. Integration is facilitated by a change of variables:

$$t = \frac{1}{z^2} \quad , \qquad \text{B5}$$

which yields the equation

$$r = \int_0^Z \frac{Qkrz}{4\pi n_e^2 \alpha^2} \exp\,(-r^2z^2/4\alpha)\; dz \qquad \text{B6}$$

The integration takes place in the reverse direction, since at $z = 0$, $t = \infty$. The initial condition for $z = 0$ for the radius is the maximum

$$r\ (t = \infty) = \sqrt{\frac{Q}{\pi n_e H}} \quad . \qquad \text{B7}$$

For the present case, the parimeters for equation B7 are

$Q = 93583\ ft^3$

$k = 28$ ft/day

$H = 25$ ft

$n_e = 0.1$

$R = kH/n_e = 7000$

The radius of the circle for the present case as a function of time is shown in figure 4.

The expanding circle travels with the ambient pore velocity. The flux of contaminant C (curies/ft^3) into an intersecting river can be approximated by the integral .

$$\text{Flux} = 2 \int_0^Y (H + h)\, u n_e\, C e^{-\lambda t}\, dy \text{ curies/day} \qquad \text{B8}$$

where

λ is the decay coefficient = $\frac{\ln 2}{\text{half life}}$,

$Y^2 = r^2 - L^2$

$L = x_o - ut$ and

h is defined by equation B2.

This integral may be evaluated analytically:

$$\text{Flux} = e^{-\lambda t}\, u\, n_e\, C \left\{ 2HY + \frac{1}{2n_e \sqrt{\pi \alpha t}} \exp\left(- \frac{\ell^2}{4\alpha t} \right) \text{erf}\left(\frac{Y}{4\alpha t} \right) \right\} \qquad \text{B9}$$

For the present case, C = 0.95 μ ci/ml and halflife = 12.33 years. The flux calculated from equation B9 is shown in figure 5. Since the travel time for the contaminated water will be hundreds of days in this case, it is safe to say that the circle of contamination has nearly reached maximum radius at the river. The approximate maximum flux is therefore

$$\text{Flux (max)} = 2Hr\,(\text{max})\, u\, n_e\, C \exp(-\lambda\, t_{max}) \qquad \text{B10}$$

where

$$t_{max} = \frac{x}{u} \quad .$$

For the present case,

$x = 600$ ft

$u = 1.7$ ft/day

$C = 0.95\ \mu$ ci/ml.

The maximum flux of tritium is therefore

$$\text{Flux (max)} = 2.8 \times 10^{-4} \text{ curies/sec} \ ,$$

which is in close agreement with the more precise equation B9.

References

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2. FSAR, Section 2.4.13, Three Mile Island, Unit 2.

3. Carslaw H. S. and Jaeger, J. C., 1959, Conduction of Heat in Solids, Oxford University Press, London.

4. Codell, Richard B. - NRC Workbook on Hydrologic Dispersion - NUREG to be published 1980, U.S. Nuclear Regulatory Commission.

5. Perkens, J. F., Merritt, W. F., and Cherry, J. A., 1977, "Studies of Dispersion in a Sandy Aquifer," Paper no. H-62 presented at AGU Spring meeting, Washington D.C., 1977.

6. McWhorter, D. B., and D. K. Sunada, Ground-Water Hydrology and Hydraulics, Water Resources Publications, Fort Collins, Colorado, 1977.

Figure 1 - Reactor Building

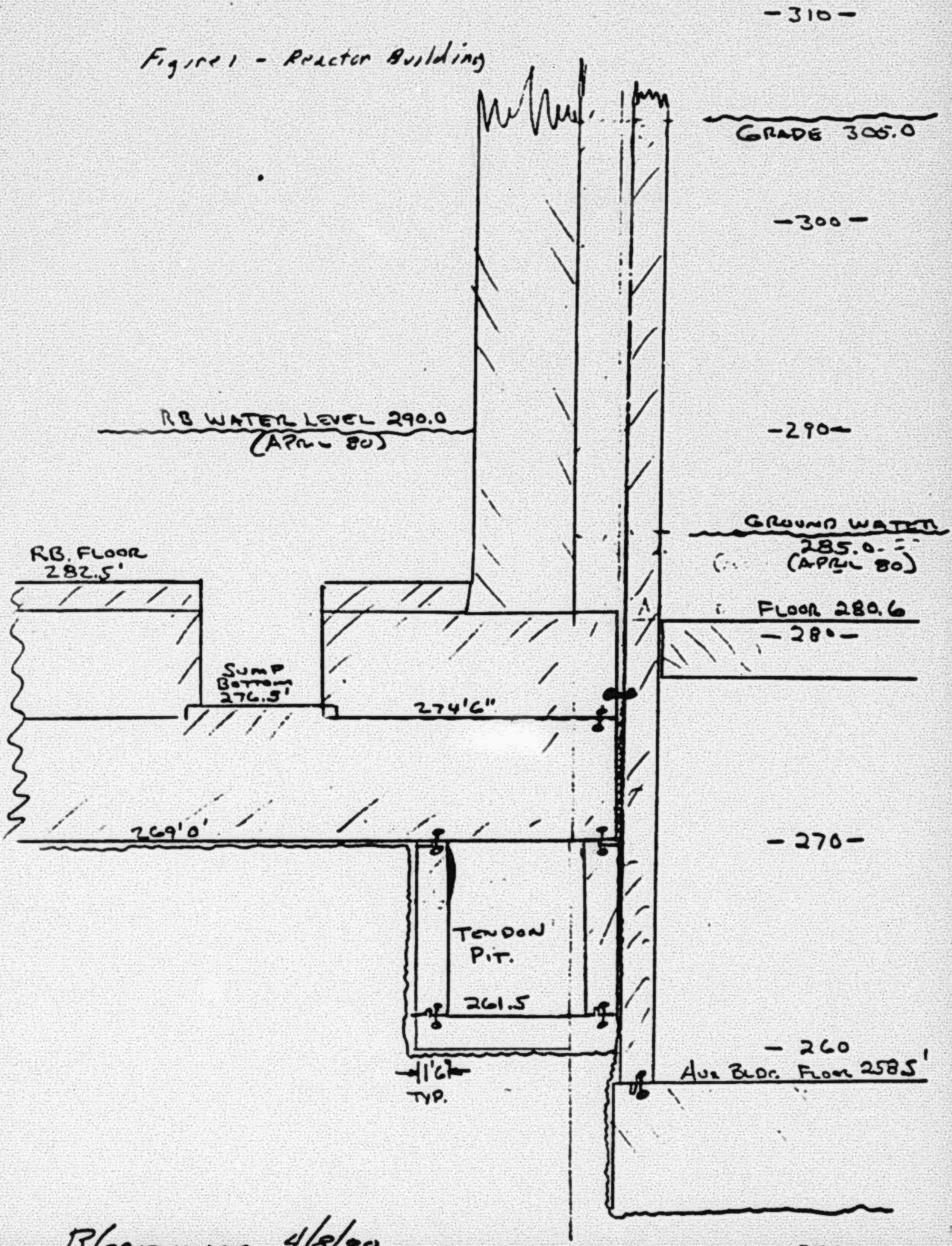

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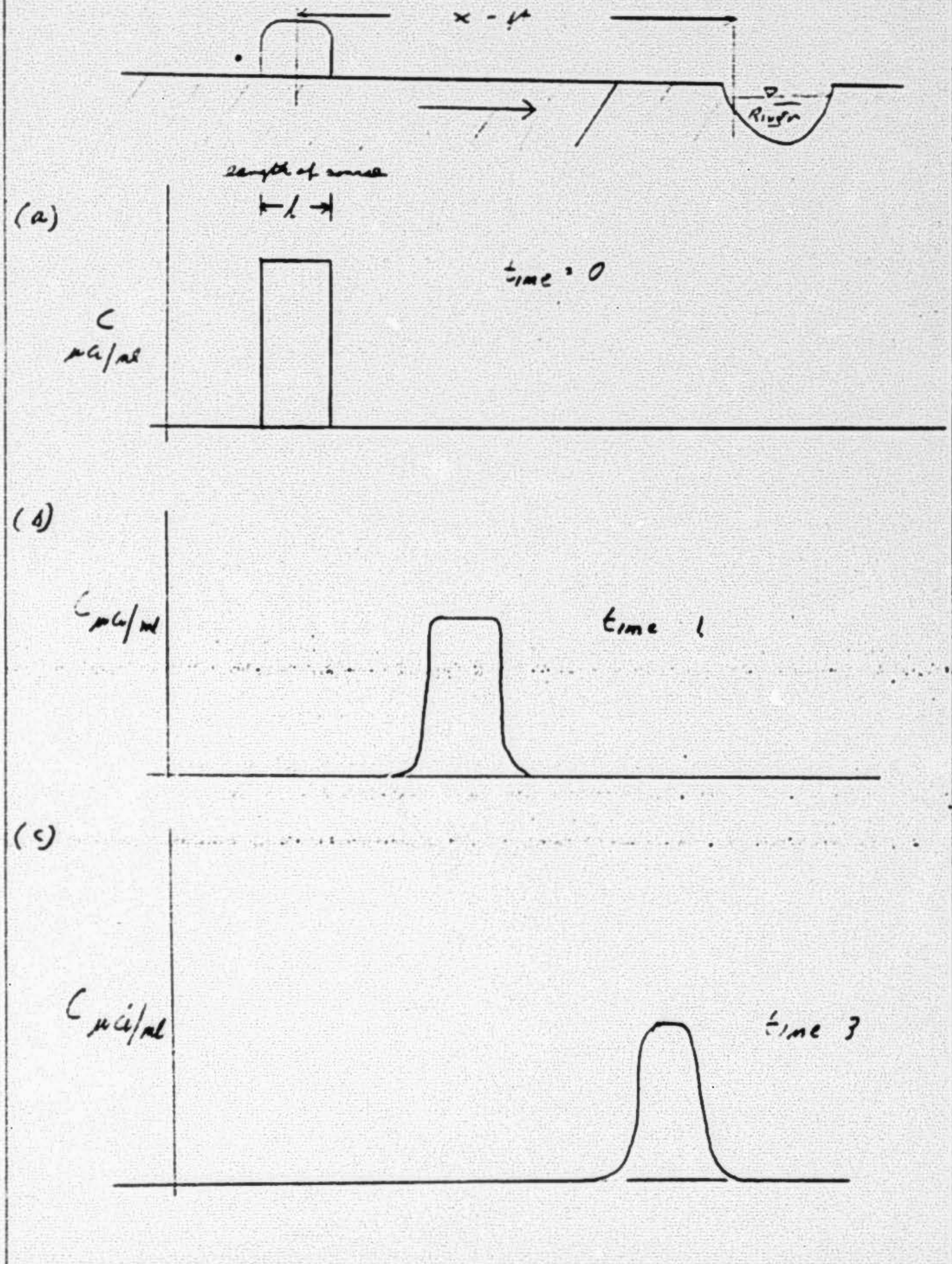
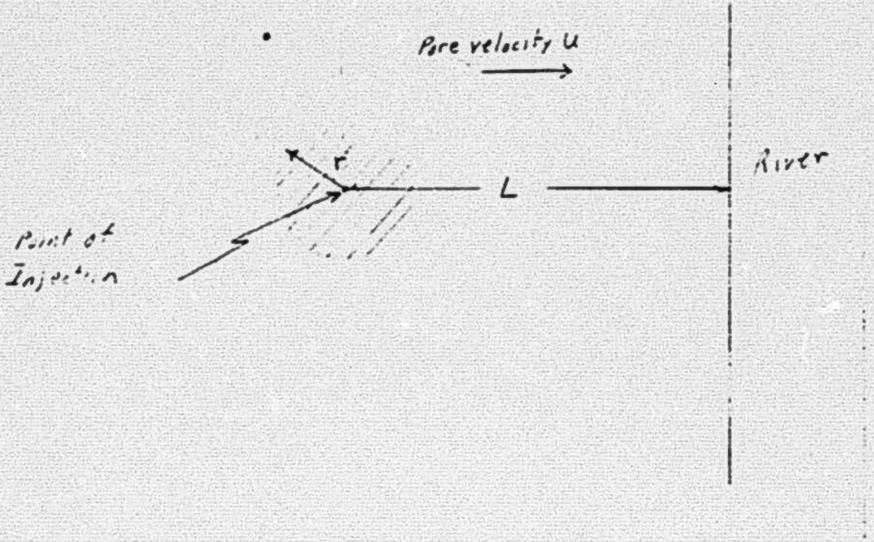
Figure 2 - Elution of Radionuclides



(a) Plan view



(b) Cross sectional view

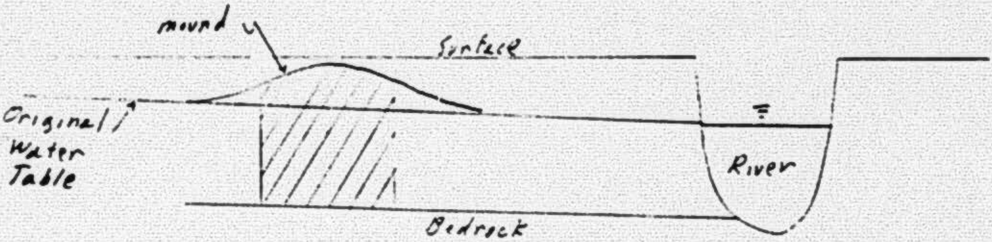


Figure 3 - Spread of Contaminated Water from Large Volume Release

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Figure 4

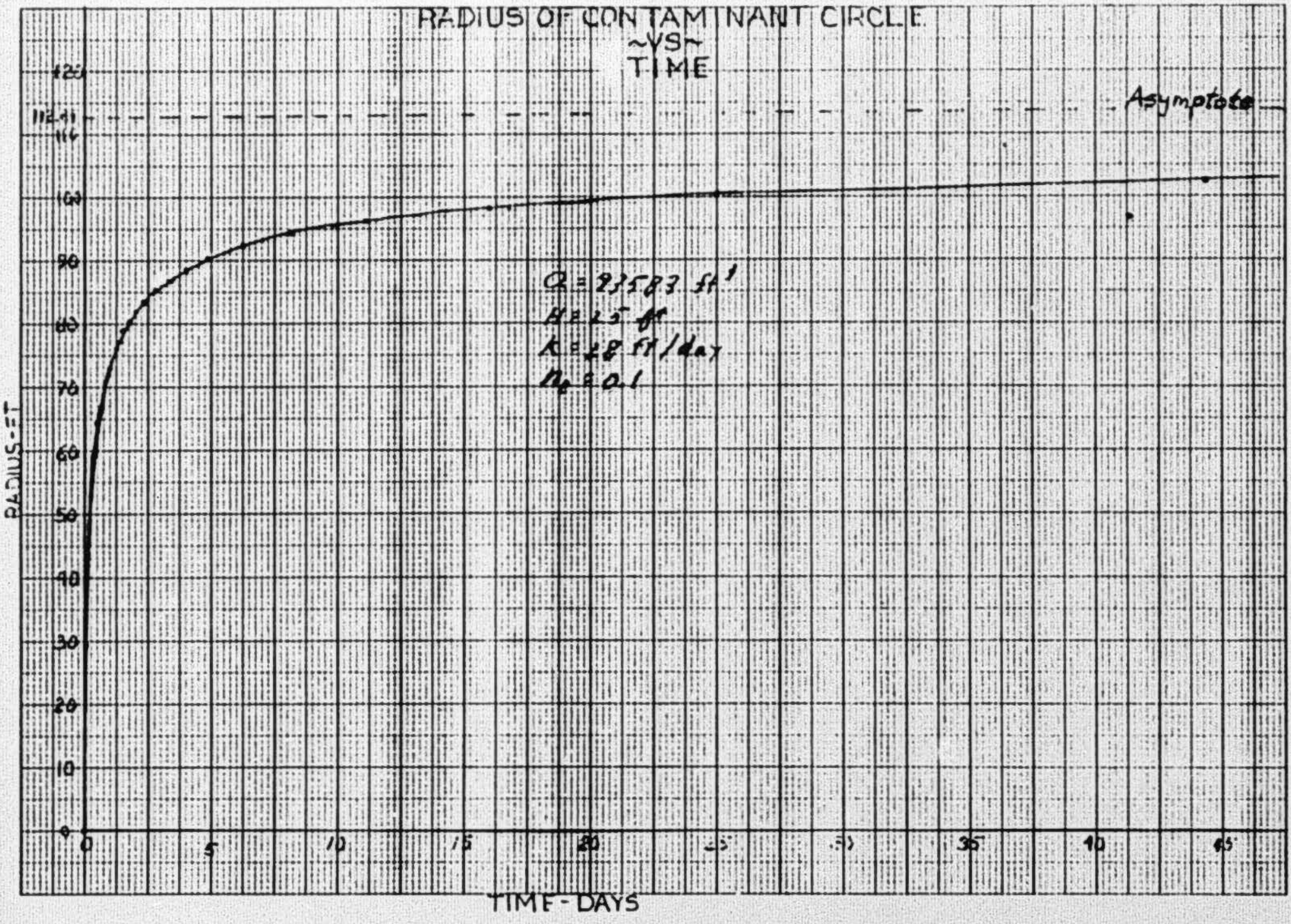

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Figure 5